FINAL

Part C:
Ecological Risk Assessment for PCB Sites
A Guide for Determining The Risk of PCB Exposure to Ecological Receptors

December 2005

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PART C: ECOLOGICAL RISK ASSESSMENT FOR PCB SITES

A GUIDE FOR DETERMINING THE RISK OF PCB EXPOSURE TO ECOLOGICAL RECEPTORS

1.0 INTRODUCTION

Polychlorinated biphenyls (PCBs) are a class of some of the most persistent chemicals in the environment and therefore require special consideration when conducting ecological risk assessments (ERAs) for ecosystems that may have been contaminated with PCBs. Made up of 209 separate compounds (or congeners) worldwide distributions of PCBs have been found in the Arctic, Antarctic, deep sea (Tilbury et al. 2002, Froescheis et al. 2000, Looser et al. 2000), as well as coastal and inland areas located close to sources of PCB contamination (Johnson et al. 2000). This guide provides basic information relevant to assessing the ecological risk of PCBs to aquatic and terrestrial ecosystems. Physicochemical information about PCBs, methods for determining PCB concentrations in water, sediment, soil, and fish and wildlife tissue samples, toxicological effects of PCBs on aquatic and terrestrial wildlife, and approaches applicable to formulating and assessing ecological risks of PCBs are reviewed. Specific information on developing ERA benchmarks for PCBs, analyzing PCB congener distributions, and current literature on evaluating the bioaccumulation and toxicity of PCBs are reviewed and links to primary literature sources are provided in this guide. Information and examples of how to effectively incorporate reference and background conditions when conducting ERAs and considerations to better coordinate ecological and human health risk assessments at Navy sites are also presented and discussed.

This guide was developed as "Part C" of "PCB ANALYSIS AND RISK ASSESSMENT AT NAVY INSTALLATIONS" to provide a framework for assessing the ecological and human health risks at Navy Sites. It incorporates contributions provided in an earlier draft prepared by Drs. Richard L. DeGrandchamp and Mace G. Barron for the Navy Environmental Health Center.

Current Navy guidance recommends using a tiered approach for conducting ERAa (CNO 1999, U.S. EPA 1992, 1998c, 2001a). First, a screening level ERA (Tier I) is conducted to determine if site contaminants are likely to pose a risk to ecological receptors. Screening level ERAs "are simplified assessments that can be conducted with limited data by assuming values for parameters for which data are lacking" (U.S. EPA 1998c). To reduce the chance of underestimating risk, conservative parameters are used in the risk assessment. Based on the outcome of the screening level assessment, a more detailed baseline risk assessment (Tier II) approach can be refined and focused on critical aspects of risk. A description of the Navy's tiered approach can be found on the Internet at http://web.ead.anl.gov/ecorisk.

1.1 Physicochemical Data for PCBs

Banned from manufacturing and distribution since 1978, polychlorinated biphenyls (PCBs) are highly bioaccumulative and the U.S. EPA has developed a strategy for protecting human health and the environment from exposure to PCBs and other persistent, bioaccumulative, and toxic (PBT) pollutants (U.S. EPA 1998a). Used extensively in the manufacturing of electrical capacitors, carbon-less copy paper, fire retardants, and other applications that required products with high heat resistance, elasticity, and durability, many PCBs have been improperly

disposed resulting in an almost ubiquitous contamination of the environment. The very properties that made PCBs so desirable for industrial applications are the same properties that cause PCBs to be resistant to degradation and to accumulate in the environment. PCBs are a mixture of compounds that consist of ten homologue groups (mono- through deca-biphenyl) and 209 different PCB congeners.

- See ***Part A:*** Overview of PCBs
- See EPA Region V web site for PCB Species Identification.
- See NAVFAC's Polychlorinated Biphenyls (PCB) Multimedia Training Tool for more information about PCB history, fate and transport, nomenclature, methods, and more.)

The physicochemical properties of PCBs govern their behavior in the environment. Key properties include solubility in water, vapor pressure, octanol-water partition coefficient (K_{OW}, also referred to as Log P), bioconcentration factor (BCF), and degradation rate. Relative to other organic compounds such as aliphatic hydrocarbons, polycyclic aromatic hydrocarbons, and nonchlorinated pesticides, PCBs have much lower solubility in water, low vapor pressure (semivolatile), higher K_{OW}, very high BCF, and very low degradation rates (MacKay, Shiu, and Ma 1992). Because PCBs are very hydrophobic (readily come out of solution), persistent, and highly lipophilic (partition into lipids and organic carbon) they readily adsorb onto particles and build up in the food chain (bio- and geoaccumulation, Froescheis et al. 2000). The concept of fugacity, or the mass transfer of a chemical from one compartment (atmosphere, hydrosphere, geosphere, or biosphere) to another as a function of its chemical properties is usually used to model the behavior of PCBs in the environment (MacKay, Shiu, and Ma 1992, Connolly et al. 2000).

PCBs have been implicated as toxic agents capable of affecting reproduction and endocrine function in birds, fish, and mammals (Johnson et al. 2000, Alston et al. 2003). Although not necessarily toxic at low concentrations, their capacity to accumulate in the environment means that organisms at higher trophic levels (higher in the food chain) are more at risk of toxic exposure to PCBs (Burreau et al. 2004, Barnthouse, Glaser, Young, 2003). Recent evidence, reviewed and documented in a peer review workshop report on PCBs, suggests that some PCBs have dioxin-like properties that can lead to carcinogenic effects in mammals including humans (U.S. EPA 1996b).

See information on physicochemical properties of PCBs for more information on obtaining physicochemical data for PCBs.

1.2 Analytical Methods for PCBs

The analytical methods selected for PCB analysis will play a very important role in making decisions about risks from PCBs (please see Part A of this guide for discussion on the various methods for PCB analysis, pros and cons, and examples of when their use may be applicable). Because of the different solubilities, volatilities, and rates of uptake, degradation, and metabolism of the individual congeners that make up the mixtures, there may be vast differences between the source materials and the PCBs observed in the environment (Van den Berg et al. 1998). Due to these effects, collectively referred to as "weathering," the use of commercial Aroclors as standards to quantify PCB levels in environmental samples has lead to

imprecise and semiquantitative results (Rushneck et al. 2004) and created problems in achieving low detection levels and the precision and accuracy needed to meet data quality objectives for ERAs (Valoppi et al. 1998, 2000). Due to weathering and environmental transformations Aroclor mixtures will seldom maintain their original composition once they are released into the environment. For example, Aroclor 1268 is composed of 87% octa- and nonachloraphenyls (homologues with 8 and 9 chlorines, respectively), which are very heavy molecules with very low solubilities and are unlikely to migrate. But the lower chlorinated components which compose only a small percentage of the original mixture would be much more susceptible to migration and (especially) bioaccumulation. Therefore the environmental fate of Aroclor mixtures, which may very well be the original source of contamination, cannot be resolved by analyzing for the parent mixture.

Better analytical methods have resulted in more accurate assessments of PCB exposure and a better understanding of the fate and transport of PCBs in the environment. Congener analysis has also been used to identify sources and mechanisms of PCB transport (Butcher and Garvey 2004) and bioaccumulation (Sather et al. 2001, Jackson et al. 2001, Hirai et al. 2004, Burreau et al. 2004). For example, the signature of PCB congeners (the relative distribution of individual congeners) was used to identify sources of ongoing PCB loading in the Hudson River from pore water diffusion and releases from contaminated sediment, which were most likely mediated by bioturbation and disturbance of the bottom sediments (Butcher and Garvey 2004). Relatively, low cost screening methods have also been developed to screen soils, non-aqueous liquid wastes, and sediments (Kirtay and Apitz 2000) for the presence of PCBs using an immunoassay (Method 4020). Low cost methods are also available to assess the ecological exposure to dioxin-like compounds by screening samples with a biomarker assay that can detect the presence of dioxin-like compounds (coplanar PCBs, PAHs, and dioxins/furans, Method 4425, Anderson and Jones 1997, Anderson et al. 1999).

In many cases, historical data sets were developed using Aroclor methods and approaches are needed that can relate congener and Aroclor results to evaluate trends and assess the effectiveness of remedies. Methods for converting and comparing sums of measured congeners (sumPCB) to total PCB (tPCB) are listed in Table C-1 The National Oceanic and Atmospheric Administration's (NOAA) Status and Trends Program routinely monitored 18 PCB congeners in sediment and tissue samples (NOAA 1991) and the same approach was used in the U.S. EPA Environmental Monitoring and Assessment Program (EMAP) conducting since the early 1990s. More recent studies have included additional dioxin-like coplanar congeners in the suite of congeners analyzed (Johnston et al. 2002). It is becoming widely recognized that conger-specific methods are superior to Aroclor methods, which usually produced substantially higher estimates of PCB concentrations (Butcher et al. 1997, Valoppi et al. 1998, 2000). As was discussed in PCB ANALYSIS AND RISK ASSESSMENT AT NAVY INSTALLATIONS Part A: Overview of PCBs choosing the appropriate analytical scheme for assessing PCBs at individual sites is very complex. Selecting the proper method will be based on site-specific considerations (Bernhard and Petron 2001) but in general, sufficiently characterizing the site upfront will avoid the cost of repeated sampling, reanalysis, and selecting inappropriate remedies.

- See PCB ANALYSIS AND RISK ASSESSMENT AT NAVY INSTALLATIONS Part A: Overview of PCBs.
- See PCB methods summary for more information on analytical methods for PCBs.

Table C-1. Source Of Data, Applicable Congeners, And Regressions For Converting PCB Congener (Sumpcb) Data Into Total PCB (Tpcb) Values.

Source of Regression and Applicability	PCB Used in the Regression	Regression Equation
NOAA Status and Trends – coastal and estuarine areas of the United States (T.P. O'Connor, personal communication)	(NOAA 18) 8, 18, 28, 44, 52, 66, 101, 105, 118, 128, 138, 153, 170, 180, 187, 195, 206, 209	Results are reported as ppb dry weight
Sediment – All Data		$tPCB = 1.98(sumPCB_{18}) + 0.97$ $r^2 = 0.992$
Sediment – Atlantic Coast		$tPCB = 1.99(sumPCB_{18}) - 1.7$ $r^2 = 0.993$
Sediment – Gulf Coast		$tPCB = 2.05(sumPCB_{18}) + 2.4$ $r^2 = 0.926$
Mollusk Tissue – Atlantic Coast		$tPCB = 1.95(sumPCB_{18}) + 3.35$
Mollusk Tissue – Gulf Coast		$tPCB = 2.165(sumPCB_{18}) + 2.82$
EMAP Carolinian and Louisianan Provinces (T.L. Wade, personal communication)	(NOAA 18) 8, 18, 28, 44, 52, 66, 101, 105, 118, 128, 138, 153, 170, 180, 187, 195, 206, 209	Results are reported as ppb dry weight
Sediment and Tissue from SE US		$tPCB = 2.19(sumPCB_{18}) + 2.19$
SINKEX – Deep Ocean Ship Disposal (Johnston et al. 2005c)	(SINKEX 26) 8, 18, 28, 44, 49, 52, 66, 77, 87, 101, 105, 118, 126, 128, 138, 153, 156, 169, 170, 180, 183, 184, 187, 195, 206, 209	Results are reported as ppb dry weight
Fish Tissue from deep ocean		$Log(tPCB) = 0.974(Log(sumPCB_{28})) + 0.41$ $r^2 = 0.93$
Coastal ecosystems in British Columbia and Northwest Territories (Sather et al. 2001)	All 209 congeners	Results are reported for total Aroclor (tAroclor) as ppb lipid (see Sather et al. 2001, Fig. 1).
Seal, sturgeon, and crab tissues		$tAroclor = 1.079(sumPCB_{209}) + 419.6$ $r^2 = 0.96$
Hudson River Fish; Translate historic Aroclor data to results comparable to modern methods (Butcher et al. 1997)	Aroclor 1016 and 1254	See reference (Butcher et al. 1997)

1.3 Formulating Ecological Risks for PCBs

Measurements or estimates of total PCB exposure are an important component of ERAs at sites with PCB contamination. The majority of available bioaccumulation factors and ecotoxicity benchmarks are based on total PCBs. These data are mostly from either dosing studies with commercial Aroclor mixtures or quantifying field effects using Aroclor-based total PCB measurements, although there is a growing body of literature of studies that have documented congener specific effects (See review of current literature on the effects of PCBs on aquatic and terrestrial ecosystems).

The first step in evaluating PCB exposure and toxicity to ecological receptors is to formulate the problem. Problem formulation will define the scope and level of conservatism of the risk assessment (e.g., Tier I or Tier II) and specify the pathways, receptors, and risk questions that will be evaluated. Products of problem formulation include a conceptual site model (Figure C-1) and a list of assessment and measurement endpoints (Table C-2) that frame how PCB exposure, toxicity, and risks will be assessed. Problem formulation will focus the ERA on the site areas with potential risk, determine what additional sampling is needed, and establish whether a more detailed investigation is warranted.

The objective of a Tier I ERA is to use conservative exposure parameters and ecotoxicity benchmarks to determine if site contaminants are likely to pose a risk to ecological receptors. To reduce the chance of underestimating risk, conservative parameters are used in Tier I ERAs (Table C-3). This results in a conservatively biased estimate of risk, which can be used to eliminate the need to conduct a more detailed risk assessment if the conservative screening evaluation results in "a defensible conclusion that no ecological risk exists" (U.S. EPA 1998c). Screening level risk assessments are recommended as an important step in developing a defensible approach for evaluating ecorisks (US EPA 1992, 1998d). Based on the outcome of the screening level assessment, a more detailed baseline risk assessment (Tier II) approach can be refined and focused on critical aspects of risk. For PCB contaminated sites, this typically involves using existing data to compare maximum concentrations of total PCBs (e.g., sediment and surface water concentrations, estimated biota concentrations) to conservative ecotoxicity benchmarks. Table C-3 summarizes the typical assumptions and considerations used in Tier I and Tier II assessments.

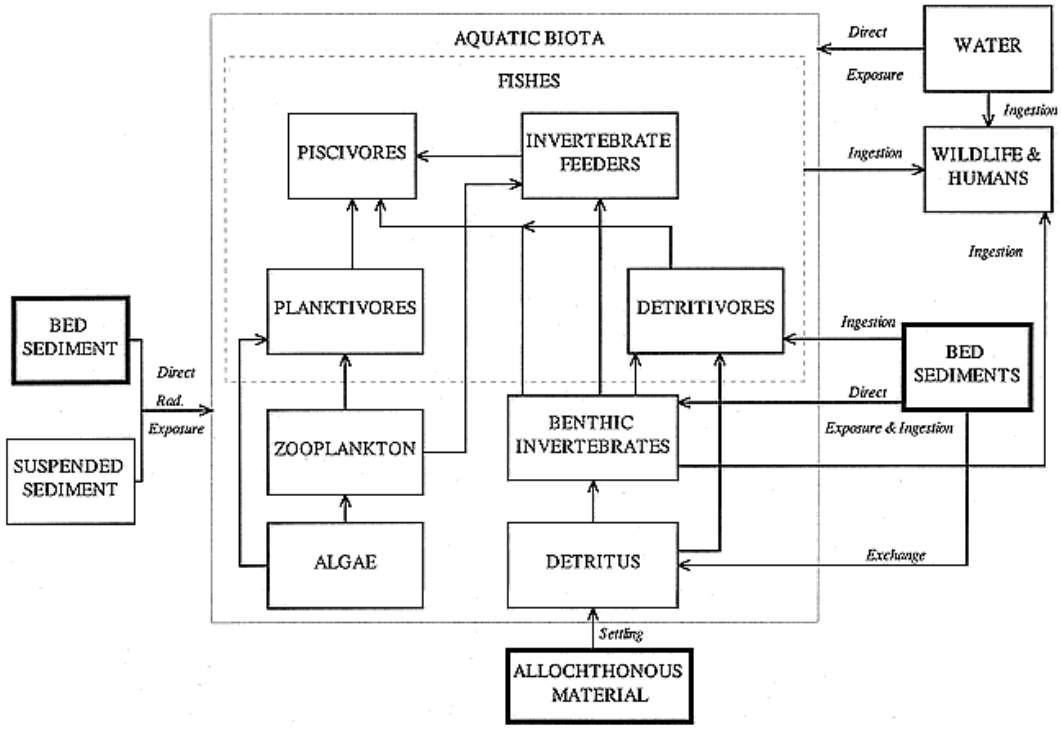

Figure C-1. Example of a conceptual model of contaminant sources, transport, and exposure in an aquatic food web. Dark-lined boxes represent sources that may be remediated. Arrows represent routes of transport and exposure, described by the labels. Unlabeled arrows represent food web transfers by ingestion. From Cook, Suter, and Sain 1998 (permission pending).

Sampling of environmental media should be performed if PCBs are suspected at the site but no data exist at the site, or the existing data show no risks but do not encompass areas of potentially higher contamination. The conceptual site model (CSM, Figure C-1) should be used to guide sampling in those areas and media with potential contamination. Based on the CSM data quality objectives (DQOs) should be developed to determine whether a complete pathway for exposure could exist. A sampling plan to assess Tier I ERAs would focus on surface soil (0 - 30 cm), surface sediment (0 – 10 cm), and surface water collected in proximity to release areas. Sediment sampling should focus on depositional areas (e.g., silty rather than sandy sediments and sediments with higher organic carbon) because these locations are likely to contain the highest concentrations of PCBs. Selecting the proper method will be based on site-specific considerations (Bernhard and Petron 2001), however, in most cases quantifying total PCB based on homologue analysis, with the option of quantifying individual congeners, is preferred over Aroclor-based analysis methods.

Table C-2. Examples Of Assessment And Measurement Endpoints And Risk Questions Addressed By Eras

Receptor Category	Assessment Endpoint	Measurement Endpoint		Risk Question
		Tier 1	Tier 2	
Plants	Survival, growth, and reproduction of phytoplankton, macro-algae, and other plants	Comparison of water concentrations of PCBs to water quality criteria	Phytotoxicity tests (e.g., lettuce, native/wild plants)	Are PCBs in surface water causing risks to aquatic or terrestrial plants?
Benthic invertebrates	Survival, growth, and reproduction of benthic invertebrate communities	Comparison of sediment concentrations of PCBs to sediment toxicity benchmarks	site-specific toxicity tests; benthic invertebrate community indices at reference and site areas	Are PCBs in sediments causing risks to benthic invertebrates?
Fish and water column invertebrates	Survival, growth, and reproduction of aquatic organisms	Comparison of surface water concentrations of PCBs to AWQC[1] and aquatic toxicity benchmarks	Comparison of fish tissue concentrations of PCBs to tissue residue benchmarks	Are site contaminants in surface water causing risks to fish and water column invertebrates?
Wildlife	Survival, growth, and reproduction of wildlife	Comparison of ingested doses of PCBs to dietary toxicity benchmarks for raccoons, mink, and kingfishers.		Are site contaminants in forage and prey causing risks to wildlife?
		NOAEL[2]	LOAEL[3]	

1. AWQC: ambient water quality criteria.
2. NOAEL: no observed adverse effect level;
3. LOAEL: lowest observed adverse effect level.

Tier I assessments should also take advantage of the availability of screening methods for PCBs (Method 4020, Kirtay and Apitz 2000, Method 4425, Anderson and Jones 1997, Anderson et al. 1999) to develop effective sampling schemes that provide adequate coverage of site media but only conduct expensive high resolution analysis on a subset of samples to verify the screening results. For example, in a study of Sinclair and Dyes Inlets for the Puget Sound Naval Shipyard, a large number of sediment samples were screened for PCBs and PAHs by immunoassay and heavy metals by XRF. A subset of samples were confirmed using more expensive analytical methods and the results were used to map contaminant concentrations throughout the Inlets and provide a basis for the Washington State Department of Ecology to delist many of the contaminants from the State of Washington's 303(d) list of impaired water bodies (Diefendorfer et al., 2003, Kohn et al., 2003, Kohn et al., 2005).

Table C-3. Typical Assumptions And Considerations Used In Estimating Wildlife Exposure And Risks In Tier 1 And Tier II Assessments[1]

Parameter	Units	Tier I	Tier II
Exposure Point Concentration (EPC)	media specific	maximum	average
Frequency of detection	NA[2]	not considered, if detected	considered in determining contaminants of concern (COCs)
Receptors	NA	most sensitive	site-specific
Body weight	kg	minimum	average
Ingestion rate	g/day	maximum	average
Prey Selection	%	most contaminated	site- and species specific
Bioavailabilty	%	100%	Modeled or measured estimates of bioavailability
Area Use Factor	unitless	1	site and species-specific based on area of home range and affected habitat
Exposure Duration	unitless	1	1 or species-specific
Bioaccumulation factor	unitless	[3]UCL literature value or 1	median literature value or site-specific measurement
Background concentrations	NA	evaluated, if available	considered in risk description and uncertainty analysis
Toxicity benchmark[4]	NA	NOAEL	LOAEL
Risk characterization	NA	Hazard quotient (HQ)[5]	Weight of evidence (e.g., HQs; toxicity testing; ecological surveys)

[1] Tier I: screening level assessment; Tier II: baseline assessment. Step 3a of Navy guidance (CNO, 1999) uses Tier II exposure assumptions and Tier I toxicity benchmarks and risk characterization approaches.
[2] NA: not applicable.
[3] Upper confidence limit (e.g., 90% UCL).
[4] NOAEL: no observed adverse effect level used as a screening value in Tier I based on sensitive receptors. LOAEL: lowest observed adverse effect level used in deriving toxicity reference value in Tier II based on likely species using the affected habitat. Tier I benchmarks should be used for special status species in Tier II assessments.
[5] HQ: EPC divided by the toxicity benchmark

If PCBs are not screened out in the Tier I assessment, a Tier II or baseline ERA usually needs to be performed. The focus of Tier II is refining the risk estimates using site-specific and species-specific exposure parameters and less conservative ecotoxicity benchmarks (Table C-3). Because PCBs are so susceptible to bioaccumulation, sampling and analysis in Tier II should include collecting co-located biological and sediment samples to provide an estimate of site-specific bioaccumulation factors or BAFs (e.g., the concentration of PCBs in benthic invertebrates normalized by lipids divided by the concentration of PCBs in sediment normalized by organic carbon, Boese et al. 1996). Biota sampling will also reduce uncertainty in wildlife exposures by using measured rather than estimated prey concentrations. This has the advantages of accounting for site-specific bioavailability of PCBs. Tier II can also include performing ecological surveys to evaluate ecological impairment relative to reference areas and bioassay

tests to determine toxicity in site media. For Tier II assessments, it is appropriate to use more precise methods such as homologue or congener analysis to better determine actual PCB concentrations and evaluate the relative distribution of congeners. Risk assessment of dioxin-like coplanar PCBs should also be considered in the weight of evidence, particularly if there is uncertainty in the risk estimates based on total PCBs. Tier II is an iterative process where additional sampling and analysis provide refined estimates of risk.

1.4 Toxicological Effects of PCBs to Aquatic and Terrestrial Wildlife

PCBs have been implicated as toxic agents capable of affecting reproduction and endocrine function in birds, fish, and mammals. Toxic responses of PCBs include dermal toxicity, immunotoxicity, carcinogenicity, and adverse effects on reproduction, development, and endocrine functions Van den Berg et al. 1998. PCBs are known to be persistent, bioaccumulative, and highly toxic to aquatic organisms and wildlife (Barron et al. 1995, Eisler and Belisle 1996). For example, PCBs can cause behavioral abnormalities, impaired reproduction, developmental toxicity, and death in birds and mammals (Barron et al. 1995; Fernie et al. 2001), and PCB exposure is associated with immune impairment, modulation of hormone levels, and tumors in fish (Barron et al., 2000).

Early toxicity studies were conducted on technical Aroclors and exposure was reported as PCB or Aroclor concentrations. In the last decade, evidence has been mounting that specific congeners are more toxic than others, especially the dioxin-like coplanar PCBs (Ahlborg et al. 1994, Van den Berg et al. 1998, Barney 2001). Dioxin-like coplanar PCBs are PCB congeners with zero or one chlorine atom in the ortho position (closest to the biphenyl double bond, see more information on orientation in Part A, and Polychlorinated Biphenyls (PCB) Multimedia Training Tool) The concentrations of these dioxin-like coplanar PCB congeners are expressed as the equivalent concentration of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD), the most potent dioxin congener (Van den Berg et al. 1998), determined from the toxicity equivalent quotient (TEQ). The TEQ is calculated by summing the products of the concentrations of individual congener [PCBcongener] and their toxicity equivalency factor (TEF):

$$TEQ = \Sigma [\text{PCB congener}_i] \times TEF_i \qquad [1]$$

Where TEF_i expresses the potency of PCB $congener_i$ relative to TCDD (i.e., TCDD TEF=1). The World Health Organization (Van den Berg et al. 1998, U.S. EPA 2000d) has established TEFs for fish, birds, and mammals that can be used in ERAs for the coplanar PCBs (Table C-4), see TEF Table on U.S. EPA PCB web site). Presently, more studies are being published in the scientific literature about the effects of PCB congeners on ecological receptors.

See review of current literature on the effects of PCBs on aquatic and terrestrial ecosystems.

Table C-4. Toxicity Equivalency Factors (Tefs) For Dioxin-Like Coplanar Pcbs And (Van Den Berg Et Al. 1998, U.S. EPA 1998e, 2000d)

PCBcongener	Chlorines[1]	Position[2]	Fish TEF	Bird TEF	Human/Mammal TEF
77	4	non-ortho	0.0001	0.05	0.0001
81	4	non-ortho	0.0005	0.1	0.0001
105	5	mono-ortho	<0.000005	0.0001	0.0001
114	5	mono-ortho	<0.000005	0.0001	0.0005
118	5	mono-ortho	<0.000005	0.00001	0.0001
123	5	mono-ortho	<0.000005	0.00001	0.0001
126	5	non-ortho	0.005	0.1	0.1
156	6	mono-ortho	<0.000005	0.0001	0.0005
157	6	mono-ortho	<0.000005	0.0001	0.0005
167	6	mono-ortho	<0.000005	0.00001	0.00001
169	6	non-ortho	0.00005	0.001	0.01
189	7	mono-ortho	<0.000005	0.00001	0.0001

[1] Number of chlorines in congener.
[2] Position of chlorine near biphenyl double bond

An assessment of dioxin-like coplanar PCB risks should be considered as part of the weight of evidence assessment in an ERA, but should not be used as the only measure of PCB toxicity (U.S. EPA 2000a). Dioxin-like coplanar PCBs are the most toxic components of the PCB mixture, but they do not account for other modes of toxicity to fish and wildlife. Also, they are very minor components of total PCB concentrations, and may require a separate and costly analysis. Because of their ability to bioaccumulate in biota and high toxicity to wildlife, analyses of dioxin-like coplanar PCBs are recommended in prey samples of wildlife. Analyses of dioxin-like coplanar PCBs in surface water, sediment, and soil are less critical because ecotoxicity benchmarks have not been well established for these media. It would be very costly to conduct chemical characterization of these media for the presence of dioxin-like compounds. However, newer, low cost, screening methods can be used to screen samples for the presence of PCBs (Method 4020, Kirtay and Apitz 2000) and dioxin-like compounds, Method 4425, Anderson and Jones 1997, Anderson et al. 1999).

2.0 DERIVING TOXICITY BENCHMARKS FOR PCBS

2.1 Benchmarks for Aquatic Ecosystems

Benchmarks should be selected to evaluate potential effects to a broad range of organisms found at the site. For aquatic ecosystems, benchmark concentrations for water (WB), sediment (SB), and tissue residues of fish (TFish) and invertebrates (TInvert) are commonly used to assess the risk of PCBs at sites suspected to be contaminated with PCBs (Johnston et al. 2005a, 2005b, MESO 2000). The available benchmarks for assessing risks of PCBs in surface water include chronic values ranging from 0.01 to 10.5 ug/L (Suter 1996) that were derived to protect fish and water column invertebrates. The U.S. EPA ECOTOX http://www.epa.gov/ecotox website contains a large database that may be useful in deriving site-specific benchmarks[1]. Current chronic AWQC for total PCBs are 0.014 ug/L (freshwater) and 0.03 ug/L (marine), which were derived for the protection of wildlife as well aquatic organisms (U.S. EPA 1998b, Buchman 1999). Currently, AWQC are not available for individual PCB congeners, or for dioxin-like coplanar PCBs. However, a body of scientific literature is growing as more studies are being published on the effects of PCBs on ecological receptors (see review of current literature on the effects of PCBs on aquatic and terrestrial ecosystems).

The available benchmarks for assessing risks of PCB contaminated sediment are based on toxicity to benthic invertebrates, and are presented as sediment toxicity screening values (mg/kg sediment dry weight) for total PCBs. Screening values for total PCBs summarized by Buchman (1999) ranged from 0.026 to 0.277 mg/kg for freshwater sediment and 0.22 to 0.19 mg/kg for marine sediment. MacDonald et al. (2000) derived consensus-based ecotoxicity benchmarks for PCBs that were considered applicable to both freshwater and marine sediments. Table C-5 lists the threshold effect concentration (TEC; 0.04 ng/g; concentration below which adverse effects are unlikely) of MacDonald et al. (2000) as the Tier I ecotoxicity benchmark, and the mid-range effect concentration (MEC; 0.4 ng/g; concentration above which adverse effects frequently occur) as the Tier II benchmark. Although sediment organic carbon is known to control PCB bioavailability, MacDonald et al. (2000) considered these benchmarks to be applicable to the range of organic carbon values that occur in marine, estuarine, and freshwater environments.

Ecotoxicity benchmarks for dioxin-like coplanar PCBs in sediment have not been established. The Canadian Ministry of the Environment (CME 2000) has proposed interim Environmental Quality Guidelines (EQGs) for polychlorinated dioxins and furans as a TEQ (i.e., equivalent concentration of TCDD). In the absence of alternative values, PCB benchmarks for Tier I (0.85 ng TEQ/kg sediment) and Tier II (21.5 ng TEQ/kg) can be used as benchmarks for dioxin-like coplanar PCBs in sediment (Table C-6).

Benchmarks of effects from PCBs for tissue residues in fish and invertebrates can also be used to assess potential ecological effects (Johnston et al 2003b, 2005a, 2005b, 2005c). The tissue residue benchmarks (Table C-7) are chemical residue thresholds at or below which adverse toxicological effects would not be expected. The benchmarks for PCB residues are based on the tissue screening value (TSV), bioaccumulation critical values (B_{CV}), critical body residues (CBR), and dietary uptake benchmarks.

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[1] To obtain the information on PCBs use the advanced query tool and select "PCBs" as a class of chemical. This will return data for all PCB compounds.

Table C-5. Examples of Tier I and Tier II Toxicity Benchmarks for total PCBs[1]

Receptor	Pathway	Tier I Benchmark	Tier II Benchmark	Source
AWQC[1]	Freshwater	0.014 ug/L	same as Tier I	EPA (1999a)
	Saltwater	0.03 ug/L		
Benthic invertebrates	sediment	0.04 mg/kg dw	0.40 mg/kg dw	MacDonald et al. (2000)
Fish[2,3]	body residue	1.9 mg/kg ww	9.3 mg/kg ww	EPA (2000a)
bird[2,4]	egg residue	2 mg/kg ww	7.6 mg/kg ww	EPA (2000a)
	ingestion	1.8 mg/kg*d	7.1 mg/kg*d	
Mammal[2,5]	ingestion (raccoon)	0.32 mg/kg*d	1.5 mg/kg*d	EPA (2000a)
	ingestion (mink)	0.004 mg/kg*d	0.04 mg/kg*d	

[1] Established guidelines are not available except for ambient water quality criteria (AWQC). Concentrations reported in either dry weight (dw) or wet weight (ww) basis; ingestion dose is mg per kg body weight (ww) per day. Tier I benchmarks should be used for special status species in Tier II assessments.
[2] Screening values are species-specific.
[3] Screening value for striped bass is listed. Residue levels will be lower for salmonids (trout, salmon).
[4] Screening value for great blue heron is listed. Ingestion value is kg (ww) body weight.
[5] Screening values for raccoon and mink are listed. Mink are among most sensitive mammalian species to PCBs and benchmarks are higher for non-related species.

Table C-6. Examples Of Toxicity Benchmarks For Dioxin-Like PCBs As Toxicity Equivalent Quotient (TEQ)[1,2]

Receptor	Pathway	Tier I Benchmark	Tier II Benchmark	Source
Benthic invertebrates	sediment	0.85 ng/kg dw[3]	21.5 ng/kg dw[3]	CME (2001)
Fish[4]	egg residue	8 ug/kg lipid	18 ug/kg lipid	EPA (2000)
Bird[5]	egg residue	0.3 ug/kg ww	0.5 ug/kg ww	EPA (2000)
	ingestion	0.0014 ug/kg×d ww	0.014 ug/kg×d ww	
Mammal[6]	ingestion (raccoon)	0.001 ug/kg×d ww	0.01 ug/kg×d ww	EPA (2000)
	ingestion (mink)	0.00008 ug/kg×d ww	0.00224 ug/kg×d ww	

[1] TEQ determined as equivalent concentration of TCDD using TEFs (see text and Table C-). Concentrations reported on a dry weight (dw), wet weight (ww), or lipid weight basis; ingestion dose is mg per kg body weight (ww) per day.
[2] Established guidelines are not available. Tier I benchmarks should be used for special status species in Tier II assessments.
[3] Screening values are interim Canadian Environmental Quality Guidelines for PCDDs and PCDFs.
[4] Screening value for striped bass is listed. Residue levels will be lower for salmonids (trout, salmon)
[5] Screening values are species-specific; screening value for great blue heron is listed.
[6] Screening values are species-specific; screening values for raccoon and mink are listed. Mink are among the most sensitive mammalian species to PCBs.

Tissue screening values (TSV), originally developed for screening-level ERAs at Navy sites (URS 2002), are the concentrations of chemicals in the tissue of an organism that would occur if water exposure levels exceeded the chronic fresh water quality criteria assuming a generic bioconcentration factor applicable to aquatic organisms (U.S. EPA 1980, URS 2002, Shepard 1998, Dyer et al. 2000). Similar in concept to the TSV, the bioaccumulation critical values (B_{CV}) for fish and invertebrates were calculated using the most recent saltwater quality criteria for chronic exposure to PCBs (U.S. EPA 1999a, Buchman 1999) and bioconcentration factors applicable to marine fish and invertebrates (Johnston et al. 2005a, 2005b).

Table C-7. Benchmarks For PCB Concentrations In Tissues Of Saltwater Fish And Invertebrates (From Johnston Et Al. 2005b)

	Residue Level mg/Kg wet weight		
Benchmark	Fish	Invertebrate	Exposure Pathway
Tier I			
TSV[1]	0.436	0.436	Effects from bioaccumulation
NOED[3]	1.500	0.600	Effects from critical body residues
$Dolphin_{NOAEL}$[5]	0.317	0.317	Effects from dietary exposure
$Gull_{NOAEL}$[5]	0.833	0.833	Effects from dietary exposure
$Cormorant_{NOAEL}$[5]	0.800	NA[7]	Effects from dietary exposure
Tier II			
Bcv[2]	0.936	7.446	Effects from bioaccumulation
LOED[4]	1.800	1.100	Effects from critical body residues
$Dolphin_{LOAEL}$[6]	1.583	1.583	Effects from dietary exposure
$Gull_{LOAEL}$[6]	8.333	8.333	Effects from dietary exposure
$Cormorant_{LOAEL}$[6]	8.000	NA[7]	Effects from dietary exposure

[1] Calculated with chronic freshwater quality criteria set to 0.014 ug/L and assuming a bioaccumulation factor (31200 L/Kg) applicable to aquatic species with an average lipid content of 3%.

[2] Calculated with chronic saltwater quality criteria set to 0.03 ug/L and assuming bioaccumulation factors applicable to marine fish and invertebrates normalized to 3% lipid.

[3] No observed effect dose

[4] Lowest observed effect dose

[5] No observed adverse effect level for dietary exposure

[6] Lowest observed effect level for dietary exposure

[7] Not Applicable to piscivore

Critical body residues (CBRs) are the tissue concentrations that, when exceeded, can cause adverse effects in an organism. The CBRs are based on bioassays where tissue residues in aquatic organisms were related to biological effects. A database of environmental residue-effects is maintained by the Army Corps of Engineers, Environmental Laboratory, Vicksburg, MS, http://www.wes.army.mil/el/ered, which can be searched to develop no observed effect dose (NOED) and lowest observed effect dose (LOED) benchmarks for ecological receptors (fish and invertebrates, Table C-7). Tissue residue benchmarks for PCBs in whole body samples of fish from the Hudson River were derived by EPA (2000a) that are applicable to both Tier I (1.9 mg/kg wet weight) and Tier II (9.3 mg/kg wet weight) (Table C-5). EPA (2000a)) also derived tissue residues for dioxin-like coplanar PCBs as TEQs in fish eggs, but not for whole body concentrations (Table C-6).

Dietary uptake benchmarks are obtained by calculating the concentration of PCBs in prey (fish) of a predator (dolphin) that corresponds to the toxicity reference value (TRV) – the dose or media concentration – that can cause an effect to the organism. Typically, the no observed adverse effect level (NOAEL) and lowest observed adverse effect level (LOAEL) are used to calculated the TRVs (U.S. Army 2000). Experimentally derived toxicity values for mammals (minks - $NOAEL_{mink}$) can be converted to effects levels for other mammals such as raccoons or dolphins ($TRV_{Dolphin}$) by scaling the dose to the ratio of body weight of the test species to the body weight of the receptor species using an empirical relationship (Sample et al. 1996):

$$TRV_{Dolphin} = NOAEL_{mink}\left(\frac{bw_{mink}}{bw_{dolphin}}\right)^{1/4} \quad [2]$$

Based on the similarity of toxicity values reported among avian species, the NOAEL and LOAEL reported for the rink necked pheasant ($NOAEL_{Pheasant}$) are assumed to be equivalent to the NOAEL and LOAEL for other avian receptors such as herring gulls and cormorants (Sample et al. 1996).

2.2 Benchmarks for Wildlife

Generally, historical toxicological studies on PCBs were performed using pure, unweathered Aroclor mixtures. In the context of assessing environmental contamination of PCBs, there is little chance that organisms are being exposed to unweathered Aroclors. Therefore, it is usually acceptable to interpret (or assume) that "total Aroclor" exposure levels (benchmarks) represent total PCB exposure. Practical procedures, that are widely accepted, are available for converting the sum of measured congeners to total PCB equivalents (see Table C-1) to assess risk to ecological receptors. Recently, toxicological studies have been conducted that relate effects more directly to environmentally relevant exposures of PCBs. See review of current literature on the effects of PCBs on aquatic and terrestrial ecosystems.

There is an extensive toxicity database for the adverse effects of PCBs in birds for both total PCBs and dioxin-like coplanar PCBs, which shows a large range in species sensitive and PCB mixture-dependent toxicity (Barron et al. 1995). Sample et al. (1996) lists total PCB benchmarks for birds ranging from 0.18 to 1.8 mg/kg×d (mg total PCBs ingested per kg body weight per day). Specific PCB ecotoxicity benchmarks for wildlife receptors were derived by EPA (2000a) for both bird egg residues (mg/kg egg) and ingested doses of PCBs. Example Tier

I and Tier II benchmarks for total PCBs (Table C-5) and coplanar PCBs as TEQs (Table C-6) are listed for the great blue heron, which occurs in a diversity of habitats.

Screening benchmarks derived for a variety of mammal species ranged from 0.005 to 4.7 (applicable to Tier I) and 0.055 to 11.7 mg/kg d^{-1} (applicable to Tier II), with mink being among the most sensitive of all tested animal species (Sample et al. 1996). Examples of Tier I and Tier II benchmarks are listed for minks and raccoons (less sensitive) for total PCBs (Table C-3) and coplanar PCBs as TEQs (Table C, U.S. EPA 2000a). These species occur in a diversity of habitats and span a range of species sensitivity in mammals. In comparison, Eisler and Belisle (1996) recommended total PCB ingestion of less than 0.1 to less than 1.3 mg/kg d^{-1} for the protection of wildlife. Efroymson et al. (1997) derived a preliminary remediation goal (PRG) for total PCBs of 0.371 mg/kg soil for protection of highly exposed terrestrial species (i.e., shrews consuming earthworms), and a surface water PRG for total PCBs of 0.0019 ug/L for protection of sensitive wildlife (i.e., river otter) consuming fish with bioaccumulated PCBs.

More information on PCB risk assessments

- ***EPA Region 4 Polyclorinated Biphenyls (PCBs) Program Ex-Oriskany Project*** (Draft PCB Disposal Approval for Ex-Oriskany Project)
- ***Hudson River PCBs Superfund Site Reassessment***
- ***Fox River PCB cleanup***
- ***USEPA New England GE/Housatonic River website***
- ***Great Lakes Binational Toxics Strategy***

2.3 Characterizing Exposure and Effects

Analyzing PCB exposure to ecological receptors involves determining the exposure point concentration (EPC) of PCBs in ecologically relevant media: surface soil (0-30 cm depth), surface sediment (0-10 cm depth), and surface water. Groundwater is usually not considered unless it is used as a surrogate for surface water or pore water in the absence of other data. Wildlife exposures are typically estimated using simple exposure models to calculate the average daily dose (D). See Table **C-8** for additional parameters for estimating exposure to blue heron and mink.

The dietary consumption benchmarks (D) of prey tissues can determined by the following relationships:

$$D = (TRV \times UF)/F \ \mu g/g \text{ (wet weight)} \quad [3]$$

Where

TRV = Toxicity Reference Value for receptor species (e.g. No Observed Adverse Effects Level – NOAEL)

UF = Uncertainty factor

F = Dietary uptake factor (g/g body weight/day)

$$= aRdL \quad [4]$$

a = Assimilation efficiency = 0.9

R = Food ingestion rate (g/g body weight/day)

= f/bw (g/g body weight day^{-1}) [5]

f = Food consumption rate g/day

bw = Body weight g

d = Fraction of diet = 1.0

L = Fraction of life span = 1.0

In the absence of published NOAELs, the lowest observed adverse effect level (LOAEL) can be used. To make LOAELs comparable to NOAELs, uncertainty factors (UF) can be applied to benchmark concentrations (U.S. EPA 1993). Generally, an uncertainty factor (UF) of 0.1 is used to convert the lowest observed adverse effect level (LOAEL) to a NOAEL, and an UF of 0.01 is used to convert a lethal dose to 50% of the population (LD_{50}) to a NOAEL (U.S. EPA 1993, U.S. Army 2000).

Table C-8. Example Ranges Of Exposure Parameter Values For The Great Blue Heron And Mink (U.S. EPA 2000a)

Parameter	Symbol	Units	Heron	Mink
Body weight	BW	kg (ww)	1.87 - 2.88	0.55 - 1.36
Ingestion rate	IRwet	kg/d (ww)	0.284 - 0.455	0.119 - 0.145
	IRdry	kg/d (dw)	0.097 - 0.108	0.042 - 1.013
Water Consumption	WI	L/d	0.089 - 0.119	0.052 - 0.131
Diet Composition	PD	%	fish: 72 - 98 AI[1]: 1 - 18 IS[1]: 0 - 4.3	fish: 18.8 - 34 AI[1]: 13.9 - 16.5 NR[1]: 49.5 - 67 IS[1]: 1
Home range	HR	kilometers	0.6 - 1.37	1.0 - 5.0
1. AI: aquatic invertebrates; NR: non-river sources; IS: incidental sediment ingestion.				

Exposure analysis in Tier I assessments are usually based on the maximum detected concentrations of PCBs and conservative exposure assumptions. If PCBs are not detected above the analytical detection limit, the concentration can be defined as one half of the analytical detection limit. Typical assumptions and considerations used in estimating wildlife exposure and risks in Tier I are presented in Table C-3, including 100% bioavailability and 100% site use. If concentrations of PCBs in wildlife prey are unavailable in a Tier I assessment, they could be estimated using literature derived bioaccumulation factors (Table C-9). Incidental soil or sediment ingestion can be important in determining PCB risks, and standard parameters can be found in Beyer et al., (1994).

Table C-9. Example Bioaccumulation Factors (BAF) Or Bioconcentration Factors (BCF) For PCBs

Pathway	BAF/BCF Range	Source
Soil to earthworm	0 - 10.4	Sample et al. 1999
Soil to small mammal	1.2	Efroymson et al. 1997
Sediment to benthic invertebrate	1 - 23.7	U.S. EPA 2000b
Sediment to fish	0.1 - 30	U.S. EPA 2000b
Water to fish	1 - 50,100	U.S. EPA 2000b
Fish to bird or mammal	0.5 - 93	U.S. EPA 2000b
Sediment to bird	4.2 - 133	U.S. EPA 2000b

Exposure analysis in Tier II (including Navy ERA step 3a; CNO 1999) uses average or upper confidence level (UCL) to define the exposure. Frequency of detection and background sources are also considered. Concentrations of PCBs in wildlife prey can either be estimated using literature derived bioaccumulation factors or determined using measured concentrations. As shown in Table C-9, BAFs and BCF can vary substantially, with BAFs of over 10 million reported for the accumulation of some planar PCBs from surface water into shellfish (U.S. EPA 2000b).

The Tier II effects analysis typically uses lowest observed adverse effect level (LOAEL) or similar benchmarks that represent concentrations above which adverse effect are likely. Tier I ecotoxicity benchmarks should be used in Tier II if special status species (endangered or threatened species) are known to be present at the site. Tier II ERAs may also involve field studies and toxicity testing of soil, sediment, or surface water to determine if aquatic organisms or wildlife are being impacted at the site. General guidance on these procedures that are applicable to PCB contaminated sites include EPA ECO UPDATES on using toxicity tests and performing field studies for ecological risk assessment (U.S. EPA 1994a, 1994b, 1994c) and bioassessment protocols and guidance for freshwater (U.S. EPA 1999b) and estuarine and coastal marine waters (U.S. EPA 2000c). Note that EPA (1994b) suggests caution on using standard toxicity testing with PCBs because of the potential for adverse effects on reproduction that is not detected in many standardized tests (e.g., 10 day sediment bioassays). Although effects from PCBs are not generally associated with direct toxicity tests, PCBs can interact with other chemicals and cause additive, synergistic, or antagonistic effects (Farrington et al. 2001). Risks in Tier II can be quantified by HQs as in Tier I. Additionally, a probabilistic assessment of PCB risks can be used because it incorporates the variability and uncertainty in exposure and toxicity, and provides directly interpretable risk descriptions for risk managers (U.S. EPA 1999c, Johnston et al. 2001).

3.0 ASSESSING REFERENCE AND BACKGROUND LEVELS OF PCBS

Ubiquitous contamination of PCBs in the environment (Tilbury et al. 2002, Froescheis et al. 2000, Looser et al. 2002, Johnson et al. 2000) makes it difficult to attribute observed PCB levels to specific sources. Concentrations of PCBs that are present within the ecological system vary greatly across large regions from the Great Lakes (Jackson et. al 2001), Hudson River and New York Bight (Barnthouse et al 2003), to California (Froescheis et al. 2000) and the Pacific Northwest (West et al. 2001). A reliable estimate of reference and background conditions will allow the "incremental risk" posed by the site to be evaluated. In addition, an explicit definition of background and reference data developed prior to the assessment can help provide a context for interpreting the results of risk investigations (Judd et al. 2003). Furthermore, Navy policy requires that background chemical levels be evaluated to determine risks associated with CERCLA sites (CNO 2004, BMI et al. 2003). Therefore, it is very important to develop information about background and reference levels of PCBs for the ecosystem being evaluated. Background concentrations of PCBs are PCBs that are present in the environment due to processes, sources, and human activities that are not related to releases that may of occurred at the site being evaluated (CNO 2004, BMI et al. 2003).

The U.S. EPA's Environmental Monitoring and Assessment Program (EMAP) national monitoring program is an important source of background data available for ERAs. One of the more advanced monitoring programs is the coastal and estuarine monitoring program. Data available from EMAP studies can provide information that can be used to evaluate contaminant trends in biota and develop an overall assessment of the environmental conditions in the various regions of the US (Figure C-2). Although EMAP is focused on coastal areas and estuaries, which can have relatively high levels of pollutants, the sample program also includes many pristine and unimpacted locations as well (Hyland et al. 1998). For example, in the ERA conducted for REEFEX (Johnston et al. 2005a) background PCB levels were estimated from data available for the Carolinian Province through the EMAP website. Specifically, tissue residue data on fish (spot — *Leiostomus xanthurus* and croaker — *Micropogonias undulatus*), white shrimp *(Penaeus setiferus*), blue crab (*Callinectes sapidus*) and oyster (*Crassostrea virginica*) were used to estimate contaminant levels present in biota within the region. Samples of these tissues were analyzed for PCBs and heavy metals. These data were used to evaluate background/regional concentrations of PCBs and metals in fish and invertebrate tissues.

- Environmental Monitoring and Assessment Program (EMAP) Data http://www.epa.gov/emap/html/data.html
- EMAP 2000 Coastal Viewer http://epamap2.epa.gov/coastal2k/viewer.htm
- Download the Draft National Coastal Conditions Report http://www.epa.gov/owow/oceans/nccr2/downloads.html
- Puget Sound Ambient Monitoring Program http://www.psat.wa.gov/Programs/PSAMP.htm

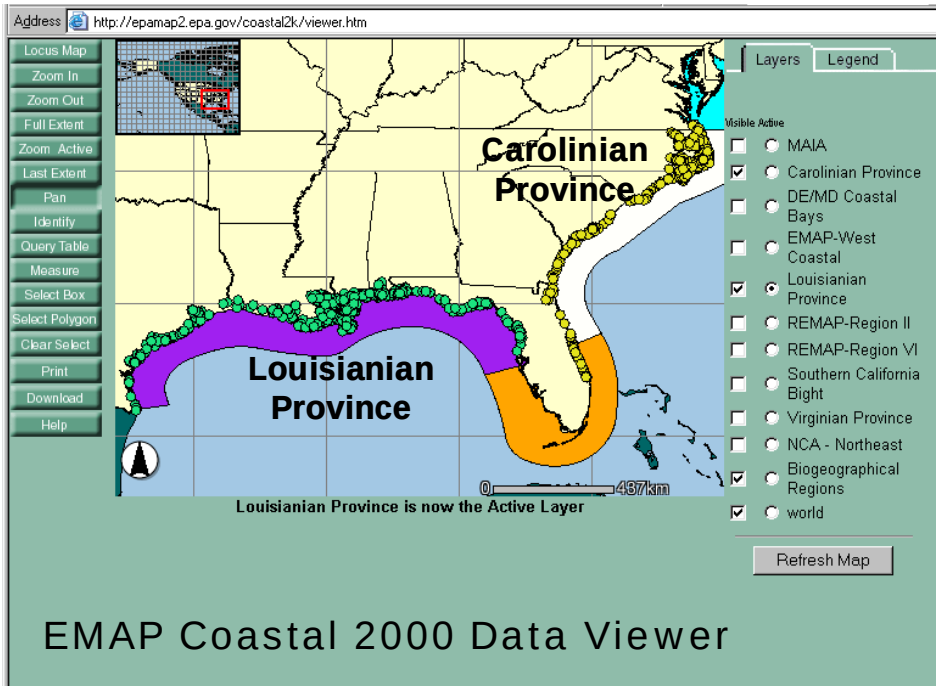

Figure C-2. Screen shot of EMAP data available for the SE US.

3.1 Coordinating Ecological and Human Health Risk Assessments

At many sites both ecological and human health risks from PCBs will need to be addressed. This is especially true in cases where there is a complete pathway in the food chain to human consumers such as hunters, fishers, or commercially and recreationally harvested products like fish and shellfish. In many cases the conceptual site model for HH and ERA will identify similar sources, uptake routes, and exposure pathways for PCBs that can be evaluated simultaneously during the risk assessment process. Careful planning and consideration of DQOs is required to obtain data from site investigations that would support both ERAs and HHRAs.

There are similar steps for conducting problem formulation, exposure assessment, and risk characterization for both HH and ERA that would benefit from an integrated approach (Staveley et al. 2000). These include identification of exposure pathways, development of exposure point concentrations in site media (water, sediment, and tissue), evaluation of "sentinel" organisms for assessing exposure, effects, and susceptibility, and sharing the results of fate, transport, and food chain modeling (Staveley et al. 2000). However, fundamental differences in the assessment endpoints and receptors (species, habitats, trophic levels, etc for ERA and sensitive human subpopulations for HHRA) and effects benchmarks (water quality criteria for ERA and RfCs, RfDs, and carcinogenicity for human health) will require different types of data and information to complete the risk assessments (Staveley et al. 2000).

For example, when conducting ERAs it is usually desirable to evaluate tissue concentrations for whole body concentrations, because predators typically consume all of their prey, while human health risk assessments rely on data from specific edible tissues (e.g. fish fillets). In general, if the tissue concentrations are normalized to lipid content, there is usually quite good agreement between PCBs measured in whole body and individual tissues, Amrhein et

al 1999). A conversion factor can be calculated to relate the ratio of tPCBs in the whole body to the fillet:

$$ConvFac = tPCB_{WB}/tPCB_{Fillet} \quad [6]$$

$$ConvFacL = (tPCB_{WB}/lipid_{WB})/(tPCB_{Fillet}/lipid_{Fillet}) \quad [7]$$

The whole body tissues are composed of internal organs with relatively higher lipids (visera, liver, and gonads) as well as tissues with low lipid content (head, tail, bones, etc). If the fillets are analyzed with "skin on," which would include the layer of fatty tissue between the skin and fillet, the differences between whole body and fillet concentration are not great (Table C-10). Of course, using data from fillets with skin on in the HHRA is only appropriate if that is how it is eaten by the local subpopulation being evaluated.

Table C-10. Comparison Of Ratio Between Whole Body:Fillet tPCB Concentrations Reported In The Literature. Fillets Were Analyzed With "Skin On"

	Whole Body : Fillet		
Fish Species	wet weight	lipid weight	Reference
Rainbow Trout	1.47	0.85	Amrhein et al 1999
Coho Salmon	1.70	0.98	Amrhein et al 1999
Coho Salmon	1.50		Stow and Carpenter 1994 cited Amrhein et al 1999
Coho Salmon	1.00		Jackson and Schindler 1996 cited in Amrhein et al 1999
Black Sea Bass	1.39	0.74	Johnston et al. 2005a
White Grunt	1.19	0.84	Johnston et al. 2005a
Vermillion Snapper	1.01	0.94	Johnston et al. 2005a

The ERA will be focused on populations, communities, complex systems, and gradients of conditions while the HRA will be focused on individuals, single species (humans), morbidity and mortality (Staveley et al. 2000). In a recent study conducted for the World Health Organization on integrating human health and ERA assessments (Suter et al. 2001) the following recommendations for improving risk assessments were made:

> "For all risk assessments, an interdisciplinary team should perform the problem formulation. It should, in consultation with the risk manager, identify the stressors and sources, select the endpoints, define the environment, and develop a conceptual model. The team should then determine whether there are linkages between the sources of stressors and potentially

significant responses of both human and ecological endpoints. If there are linkages to both types of endpoint receptors, the team should then plan and carry out an assessment that uses consistent data, exposure and effects models, and risk characterization. They should communicate their results in a consistent and integrated manner so that risk managers and stakeholders understand the implications of alternative actions. Finally, they should support the decision-making process by providing results in terms that are appropriate to the decision logic used by the risk manager" (Suter et al. 2001).

4.0 SUMMARY

This guide provides basic information relevant to assessing the ecological risk of PCBs to aquatic and terrestrial ecosystems. Physicochemical information about PCBs, methods for determining PCB concentrations in water, sediment, soil, and fish and wildlife tissue samples, toxicological effects of PCBs on aquatic and terrestrial wildlife, and approaches applicable to formulating and assessing ecological risks of PCBs are reviewed. Specific information on developing ERA benchmarks for PCBs, analyzing PCB congener distributions, and current literature on evaluating the bioaccumulation and toxicity of PCBs are reviewed and links to primary literature sources are provided in this guide. Information and examples of how to effectively incorporate reference and background conditions when conducting ERA and considerations to better coordinate ecological and human health risk assessments at Navy sites are also presented and discussed.

APPENDIX A

REFERENCES

This section contains an expanded bibliographic citation for the references listed in this report as well as additional references pertinent to PCBs in the environment. If available, the world wide web links to the Abstract, Full Text (in HTML or PDF format), and supporting information are provided. In the case of scientific journals and copyrighted materials, a subscription or membership is required to access some of the online information (usually full text and print versions of the articles). For ease of access, reprints of selected articles and reports are provided on the distribution CD and can be accessed by selecting the <Reprint> link, located at the end of the citation. Available reprints can be viewed by browsing the reference document directory on the CD (.\RefDocs).

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APPENDICES

Information on physicochemical properties of PCBs

PCB methods summary

A Review of Current Literature on the Effects of PCBs on the Aquatic and Terrestrial Wildlife